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Life-cycle carbon budget of China's harvested wood products in 1900–2015

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ABSTRACT

Harvested wood products (HWP) are essential carbon pool of the forestry sector and can play an important role in climate change mitigation. Since the 10th Conference of Parties to the United Nations Framework Conversion on Climate Change in 2009, HWP have been required to be included in forest management as an additional carbon pool in national greenhouse gas (GHG) inventory reports. China is the largest consumer of end-use HWP and the largest importer of primary HWP in the world. Thus, the carbon stocks and emissions of China's HWP must be accurately assessed for evaluating their contribution to the climate change mitigation potential of global HWP. We developed an analysis framework by using Stock-Change Approach to estimate China's HWP carbon stocks/emissions from a life-cycle perspective, which covers three HWP life-cycle stages, namely, manufacture, end use, and end-of-life disposal. Using this framework, we produced the first-ever life-cycle carbon budget for HWP consumed by China in the period 1900–2015, based on a comprehensive analysis of China-specific data. From 1900 to 2015, total wood fiber input to the HWP carbon system in China was estimated to be 8049 teragrams of carbon dioxide-equivalent (Tg CO_2-eq). Of the total wood fiber input, discarded mill residue, in-use HWP, and HWP disposed of at solid waste disposal sites retained 513, 3284, and 659 Tg CO_2-eq of carbon, respectively, with the remaining 3591 Tg CO_2-eq of wood carbon released back to the atmosphere. Of the total emission, combustion and decomposition of mill residue, burning of fuelwood, and combustion and decomposition of retired HWP accounted for 36.5%, 24.6%, and 38.9%, respectively. Landfill methane emissions were estimated to be 780 Tg CO_2-eq after considering the global warming potential of this greenhouse gas. The net HWP carbon stocks, the sum of the carbon retained by discarded mill residue and the carbon stocks of HWP in use and in landfills, minus landfill methane emissions, increased from 297 Tg CO_2-eq in 1950 to 4456 Tg CO_2-eq in 2015.

1. Introduction

Climate change is a major environmental problem that must be addressed worldwide (IPCC, 2014a). Forests are the largest carbon pool among land ecosystems (FAO, 2010) and store 861 ± 66 petagram carbon (Pan et al., 2011). Enhancing forest carbon sequestration is considered as an important and economical approach to mitigate climate change (IPCC, 2014b). Since the 10th Conference of Parties to the United Nations Framework Convention on Climate Change (UNFCCC) in 2009, harvested wood products (HWP) have been required to be included in forest management as an additional carbon pool in the national greenhouse gas (GHG) inventory report because of its potentially significant contribution in either increasing or reducing atmospheric GHG emissions (UNFCCC, 2009).

China is one of the largest HWP producers and consumers in the world (Buongiorno et al., 2017; Zhang et al., 2015). Hence, it is critical to accurately assess China's HWP carbon emissions/stocks for improving the assessment of global HWP carbon budget. Published China-specific studies mostly focused on estimating the carbon stocks of in-use HWP (Ji et al., 2013; Yang et al., 2014; Yang and Zhang, 2016), and ignored wood carbon balance during HWP manufacturing and HWP carbon stocks/emissions after retired HWP are disposed of (Brunet-Navarro et al., 2016; Ji et al., 2013; Yang et al., 2014). In the present study, we attempted to improve the analysis by developing an analysis framework that quantifies wood biogenic carbon stocks and emissions in the period from 1900 to 2015; a life-cycle perspective was used to have HWP manufacture, end use, and end-of-life disposal life-cycle stages all considered.

Winjum et al. (1998) and Lim et al. (1999) summarized three approaches, namely, Atmospheric Flow Approach (AFA), Production

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Approach (PA), and Stock-Change Approach (SCA), to estimate HWP carbon stocks and emissions. These approaches were later adopted by the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 2003, 2006a), and were then widely used in various studies (Brunet-Navarro et al., 2016; Yang and Zhang, 2016). Using PA, a reporting country should be responsible for HWP produced by domestic harvests, i.e., the carbon stocks and emissions of imported HWP are excluded from the accounting system, but those of the exported HWP are included. Hence using PA can lead to the largest HWP carbon stocks for a country that has a large net HWP export, compared with using the other 2 approaches (Donlan et al., 2012; Tonosaki, 2009). Using SCA, carbon stocks and emissions are estimated for all HWP consumed by the reporting country, regardless of where the forests are harvested to produce those HWP. In other words, imported HWPs are included and exported HWPs are excluded; therefore, using SCA will result in the largest HWP carbon stocks for a country with a large net HWP import (Green et al., 2006; Skog, 2008). The AFA estimates fluxes of carbon to and from the atmosphere for HWP consumed by a reporting country. All HWP consumed within the national boundary of the reporting country are considered, including imported HWP, while emissions from the exported HWP are excluded. Thus, using AFA will give vast emission removals for a country with large net HWP export (Dias et al., 2009). The above analyses suggest HWP trade is the main factor that affects HWP carbon assessment for a reporting country, and thus influences the selection of HWP carbon accounting approaches (Dias et al., 2009; Donlan et al., 2012; Green et al., 2006; Kohlmaier et al., 2007; Skog, 2008). The PA was recommended by IPCC as the default HWP carbon accounting approach (UNFCCC, 2012; IPCC, 2014c). Using this approach, HWP carbon stock/emission analysis can be integrated with the carbon balance assessment of the forests managed to produce these HWP in a reporting country, which can also ensure that HWP originated from deforestation and forest degradation are excluded from the analysis (Chen et al., 2010; Nunery and Keeton, 2010; Profft et al., 2009; Werner et al., 2010).

However, HWP consuming countries play a critical role in enhancing climate change mitigation potential of HWP. A wood material importer can contribute to increased HWP carbon stocks by maximizing the utilization efficiency of converting industrial roundwood or intermediate HWP into finished HWP (Hashimoto, 2008; Nabuurs and Sikkema, 2001; Werner et al., 2010; White et al., 2005). Using more HWP for long-lived end uses and increasing the recycling and reuse of retired HWP can further increase HWP carbon stocks and reduce emissions (Skog, 2008; Tonosaki, 2009; Dias et al., 2012; Chen et al., 2014). Using more HWP to substitute for carbon-intensive products, such as cement and steel, results in increased reduced GHG emissions (Chen et al., 2018; Kalt et al., 2016; Sathre and O'Connor, 2010). And waste management of the HWP consuming countries determines the fate of the HWP carbon after the HWP reaching the end of their service lives (Chen et al., 2014; Heath et al., 2011; Tonn and Marland, 2007). In summary, using PA—a HWP producing country-focused approach, the potential contribution of HWP consuming countries in enhancing HWP carbon stocks and reducing emissions can be largely underestimated, which may lead to missed opportunities in mitigating GHG emissions. The international intra-industry trade of HWP, a critical part of global forestry industries (Dieter, 2009), can result in additional uncertainty in HWP carbon assessment conducted using the PA.

China has become one of the largest HWP producers and end users in the world, and has played a complicated rule by importing and exporting large amounts of HWP. Therefore, the SCA is likely more appropriate in assessing HWP carbon stocks and emissions for China, since it is a consumption-focused approach that has the simplest data requirements and the lowest uncertainty risk (Bache-Andreassen, 2009).

Ji et al. (2013), Yang et al. (2014), and Yang and Zhang (2016) estimated the HWP carbon stocks in China: HWP production and trade data by major HWP category obtained from FAOSTAT database (http://www.fao.org/faostat/en/#data/FO). Using SCA, Yang and Zhang (2016) estimated that in 2015, the cumulative carbon stock and annual carbon stock increase of in-use HWP in China were 680.63 and 59.63 teragrams of carbon (Tg C); if PA was used, in comparison, the cumulative carbon stock and the annual carbon stock increase of in-use HWP in China were estimated to be 398.87 and 23.21 Tg C in 2015. However, these results are based on HWP production and trade data by major HWP category obtained from FAOSTAT database (http://www.fao.org/faostat/en/#data/FO), and only included carbon stocks of HWP in use (i.e., the wood carbon balance during HWP manufacturing and the fate of retired HWP were not considered). The FAO database often contains estimates made by FAO, while using country-specific datasets can improve HWP carbon assessment (Chen et al., 2018; Dias et al., 2009; Kayo et al., 2014; Skog, 2008). Lun et al. (2012) quantified the carbon budget of China's forestry sector from 1999 to 2008. However, the HWP they analyzed was limited to industrial roundwood rather than broke down to end-use HWP and end-of-life disposal. And the carbon stock was estimated using a simple decay method (Skog et al., 2004) rather than the first-order decay method recommended by IPCC (Pingoud and Wagner, 2006; IPCC, 2014c). Thus, none of these previous studies provided results comparable to those estimated by using a life-cycle perspective.

The objectives of this study include (a) to develop a SCA-based HWP life-cycle carbon stock/emission assessment system based on China-specific data on HWP production, trade, and end uses, and end-of-life disposal, and (b) to produce a life-cycle carbon budget for HWP consumed by China from 1900 to 2015. We aimed to produce a complete HWP carbon stocks/emissions assessment for the first time for China, by considering the 3 HWP life-cycle stages (production, end use, and pose-service disposal) and covering a period from 1900 to 2015.

2. Methodology and materials

To estimate the carbon stocks/emissions for HWP consumed in China using SCA, we obtained China-specific HWP production and trade data by major HWP category. A life-cycle carbon budget analysis system was developed based on a comprehensive analysis of the available data. Methods for quantifying the biogenic carbon stocks/emissions for HWP consumed by China were selected/developed for HWP manufacture, and the subsequent end use and end-of-life disposal life-cycle stages. The production of HWP by HWP category was estimated based on domestic forest harvesting, the net import of roundwood and immediate HWP, and the recycled HWP. The carbon stocks of HWP in use was estimated by HWP category, since we did not have China-specific HWP end uses and the associated service lives for conducting a higher-tier analysis. Carbon dynamics of post-service HWP were also quantified by waste HWP disposal option. The results were presented as cumulative biogenic carbon stocks of HWP in use and at waste disposal sites, minus landfill methane emissions produced by HWP decomposition in landfills. All carbon stocks and emissions in this study were presented as CO_2-equivalent (CO_2-eq), unless otherwise specified. Converting from carbon to CO_2-eq is done by multiplying the molar ratio of CO_2 and carbon (44/12), and converting methane into CO_2-eq was done by considering the molar ratio of CO_2 and methane and the 28-time global warming potential of methane in a 100-year period (IPCC, 2014b).

2.1. Life-cycle carbon budget assessment system

A complete HWP life cycle includes three stages: manufacture, end use, and end-of-life disposal (Chen et al., 2014; Lamers et al., 2014; Mathieu et al., 2011; Skog, 2008). Conducting a national level HWP life-cycle carbon budget assessment relies on a comprehensive analysis of HWP life-cycle data. The primary data that we used include the production and net imports of HWP by major product type, and the fractional shares of waste HWP discarded by disposal options. We obtained these HWP production and trade data from China Forestry

Statistical Yearbooks for the period 1950–2015 (SFA, 2016) and Almanacs of China Paper Industry from 1986 to 2015 (CTAPI, 2016), supplemented by trade data from FAOSTAT database for a few of HWP categories when trade data are missing in these two yearbooks.

Our analysis mainly focused on carbon stocks/emissions in the period from 1950 to 2015 due to data availability. But to make valid estimates for this recent period, the IPCC recommended including carbon inputs to and outputs from HWP carbon pool for HWP consumed since 1900, while assuming HWP placed in use prior to 1900 were insignificant to the current year HWP carbon stocks/emissions and thus ignored (IPCC, 2006a). Because large data gaps in HWP production and trade exist over the period 1900–1950, we used the inverse calculation method recommended by IPCC to estimate the production and trade of a HWP type from 1900 to the year when the HWP was first reported (IPCC, 2006a) (Appendix B). To estimate the missing HWP production or trade data for those years between two reported years, we assumed that the annual HWP production, import, and export changed at a constant rate.

Based on available China-specific data, we developed a conceptual model to illustrate the processes and components included to estimate the life-cycle carbon budget for HWP consumed by China (Fig. 1). In the HWP life-cycle system, the total carbon inflow to the HWP carbon pool is the total carbon contained by the net imports of finished HWP and the raw wood materials, with the latter including domestically produced roundwood, logging residue collected, net import of industrial roundwood, and net imports of intermediate HWP (wood chips, veneer sheets, and wood pulp). In particular, some logging residue, rather than mill residue, is collected in China to produce fiberboard and particleboard (SFA, 2000). To convert from volume of wood to carbon, we used the carbon conversion factors listed in Table A.1 (Appendix A). The life-cycle HWP carbon stocks and emissions comprise three components: waste wood produced during HWP manufacture that is discarded at industrial wood waste disposal sites, in-use HWP, and retired HWP disposed of at municipal solid waste disposal sites. The life-cycle carbon stocks and emissions change dynamically, with a portion of the inflow carbon stored by in-use HWP or wood and waste HWP discarded at solid waste disposal sites, and the rest released to the atmosphere through decomposition and combustion.

Some studies defined HWP end uses such as making furniture, and used as construction and packaging materials (Chen et al., 2014; Dias et al., 2012; Kayo et al., 2014; Skog, 2008), following IPCC tier-3 method supported by available country-specific HWP life-cycle data and methodologies. However, in this study, we used the tier-2 method after considering the available China-specific data (IPCC, 2006a), in which the carbon stocks of HWP in use were estimated by major HWP types. We categorized finished HWP into sawnwood, plywood, fiberboard, particleboard, other wood-based panels, and paper and paperboard in this study, which was determined by the available HWP type-

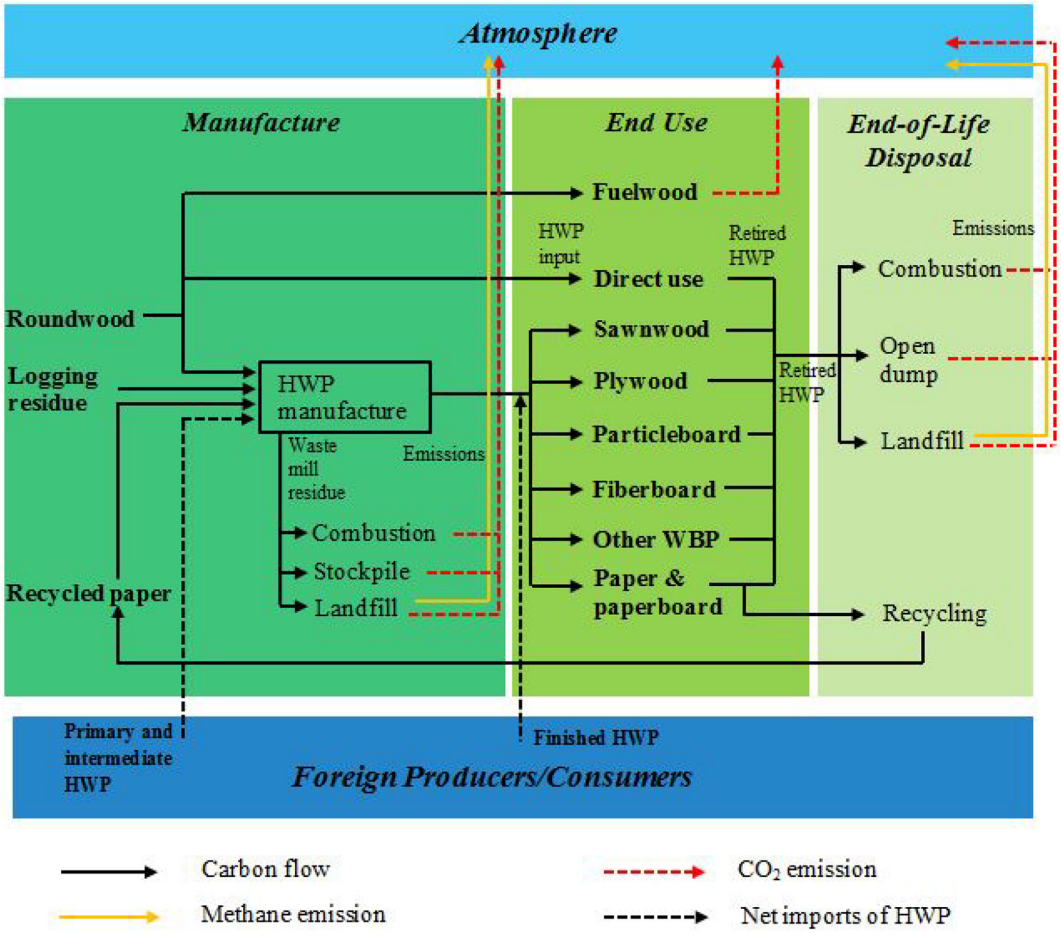

Fig. 1. Conceptual model of the analysis system designed to quantify the biogenic carbon stocks/emissions for the harvested wood products (HWP) consumed by China. The analysis covers 3 HWP life-cycle stages: manufacture, end use, and end-of-life disposal. The primary and intermediate HWP include industrial roundwood, recycled paper, wood chips, wood pulp, and veneer sheets; the finished HWP include sawnwood, plywood, particleboard, fiberboard, other wood based-panels, and paper and paperboard. Direct use: Roundwood that are used without any additional industrial process, such as building fence and using roundwood as wooden beams in wood house construction in a few rural areas in China; WBP: wood-based panels.

specific data to support the life-cycle carbon budget assessment. As described before, fiberboard and particleboard are produced using harvest residue collected as the input wood fiber. Because there is no logging residue collection data available in China, we reversely calculated logging residue collected by dividing the total production of fiberboard and particleboard by the production efficiency of these products, 88.9% (Meil et al., 2009). Other types of HWP are mainly made from domestically produced industrial roundwood, supplemented by net imports of industrial roundwood and intermediate HWP. Most of the mill residue was utilized to produce paper and paperboard, while unutilized mill residue was discarded as industrial wood waste (SFA, 2016; CTAPI, 2016). We estimated unutilized mill residue production by subtracting carbon contained by domestically produced finished HWP from carbon contained by wood fiber input to the manufacturing processes of these HWP. Some of the rounwood is harvested as fuelwood, and is burned to produce energy. And a fraction of the roundwood is directly used for applications such as log cabin or fence constructions in rural areas (hereinafter referred as direct uses of roundwood).

Retired HWP are burned without energy recovery, or discarded in landfills or open dumps, where a portion of the HWP carbon is stored and the remaining is released through decomposition. A portion of waste paper and paper products are recycled and used for pulp and paper production. The data on domestically recycled paper was obtained from Almanacs of China Paper Industry (CTAPI, 2016). The fractional shares of waste HWP disposed of by disposal option (combustion, open dump, and landfill) from 2008 to 2015 were obtained from China Circular Economy Yearbooks (Zhang, 2016). And for the period prior to 2008, these disposal fractions were estimated based on analysis in Cai et al. (2013), which is described in Section 2.4.

2.2. Carbon stocks and emissions during harvested wood products manufacturing

The first HWP life-cycle stage is product manufacturing, during which the input wood fibers are converted into finished HWP. We included domestically recycled paper as wood fiber input for estimating paper and paperboard production (CTAPI, 2016). However, the domestically recycled paper cannot be included into the annual raw wood materials inflow to China's HWP carbon pool to avoid double-counting, because the retired HWP are a part of the existing HWP carbon pool. During HWP manufacturing, a large fraction of input wood fiber is converted to mill residue, and thus the analysis of the use and disposal of these waste wood materials is important for estimating carbon stocks and emissions during HWP production. According to China Forestry Statistical Yearbooks (SFA, 2016), most of the mill residue was not utilized until the 1980s, when the booming pulp and paper industries started to use mill residue as a wood fiber source. However, the data on the fate of unutilized mill residue is not available. Thus in this study, we assumed that the unutilized mill residues were either burned, or disposed of in landfills and stockpiles using the same disposal fractions as that of retired HWP, with the uncertainty of the simplification analyzed later in the Discussion. Waste mill residue in stockpiles decomposes under aerobic conditions, producing primary CO_2. Industrial landfills create partial anaerobic decomposition conditions, with 80% of the buried waste decomposes anaerobically and produce methane and CO_2, and the rest decomposes aerobically producing CO_2 only (Chen et al., 2014; IPCC, 2006b; Skog, 2008). More details on the methods used to estimate carbon stocks/emissions of mill residue discarded in stockpiles and landfills are provided in Section 2.4. Since 2009, the domestically produced mill residue was almost completely consumed by HWP industry in China, resulting in large amounts of wood chips imported (SFA, 2009). To meet the increasing demand for wood fiber, the fiberboard and particleboard manufacturing industries in China, which also developed rapidly since 2009, have used logging residue rather than mill residue as the main input wood materials (SFA, 2009).

Production efficiency, defined as the ratio of the carbon contained by finished HWP to the carbon contained by wood fiber input to the production process of the HWP, is calculated using Eq. (1):

$$F_{I-O}(t) = \frac{C_{PF}(t)}{C_{IndR}(t) + C_{LResidues}(t) + C_{Recycle}(t) + C_{Interm}(t)} \quad (1)$$

where t is the year; $F_{I-O}(t)$ is the production efficiency; $C_{PF}(t)$ is the carbon stored in the domestically produced finished HWP; $C_{IndR}(t)$, $C_{LResidue}(t)$, and $C_{Recycle}(t)$ are the carbon contained by industrial roundwood, logging residue, and recycled paper consumed in year t, respectively, calculated as the sum of production and import, minus export of these materials; and $C_{Interm}(t)$ is the carbon stored in the net imports of the intermediate products (import minus export).

The carbon contained by wasted wood materials during HWP manufacture, $CW_{Manufacture}(t)$, can be calculated using Eq. (2):

$$CW_{Manufacture}(t) = C_{IndR}(t) + C_{LResidue}(t) + C_{Recycle}(t) + C_{Interm}(t) - C_{PF}(t) \quad (2)$$

The Eq. (1) is used to calculate the overall production efficiency of all the finished HWP. As mentioned before, the production efficiency of particleboard and fiberboard which consume logging residue collected is 0.889. For other HWP consuming industrial roundwood and recycled paper, our primary analysis showed a greater than 1.0 production efficiency from 2011 to 2015, likely caused by the underestimated production of industrial roundwood obtained from China Forestry Statistical Yearbooks (pers. comm., Kenneth E. Skog, USDA Forest Service; Joseph Buongiorno and Shushuai Zhu, University of Wisconsin-Madison). Our primary analysis suggested that in 2007–2009, the production efficiency of these finished HWP stabilized at around 0.9, and was ranked among the highest production efficiency in the world (Buongiorno and Zhu, 2015). Thus, we assumed that the production efficiency of the finished HWP except for fiberboard and particleboard remained unchanged since 2009, because it is difficult to further improve a production efficiency at this high level (Chen et al., 2014; Heath et al., 2011). Using this production efficiency, we reversely calculated the industrial roundwood production in China from 2010 to 2015 to replace the data obtained from China Forestry Statistical Yearbooks.

2.3. Carbon stocks and emissions of harvested wood products in use

Carbon contained by fuelwood is immediately released back to the atmosphere at the time when the wood is burned. Other HWP store the carbon they contain for various periods of time, determined by the service lives of product end uses. To estimate the carbon stocks of HWP in use, the IPCC recommended the first-order decay method described using Eq. (3) (IPCC, 2014c):

$$C_{Use}(t) = e^{-k} \times C_{Use}(t-1) + \left[\frac{1-e^{-k}}{k}\right] \times Inflow(t-1) \quad (3)$$

where $C_{Use}(t)$ is the cumulative carbon stock of in-use HWP in year t; $Inflow(t-1)$ is the newly consumed HWP in the year $t-1$, calculated as the sum of domestic HWP production and HWP import, minus export; $k = ln(2)/HL$ is the constant decay rate, where HL is the half-life of a HWP in an end use, defined as the time required when half of the HWP are out of service (Table C.1, Appendix C).

2.4. Post-service harvested wood products carbon stocks and emissions

Harvested wood products eventually retired from their end uses. The carbon stored by the retired HWP within year $t-1$, $C_{Discard}(t)$, can be calculated using Eq. (4):

$$C_{Discard}(t) = [C_{Use}(t-1) + Inflow(t-1)] - C_{Use}(t) \quad (4)$$

where other variables are defined as the same as for Eq. (3).

According to the Circular Economy Yearbook of China (Zhang,

Table 1
Periodic carbon balance (Tg CO_2-eq) during harvested wood products (HWP) manufacturing in China. The production efficiencies are the periodic averages of converting virgin wood fiber and domestically recycled paper to finished HWP.

Period	Virgin wood fiber consumption[a]		Domestically recycled paper	Production of finished HWP				Net import of finished HWP	Industrial wood waste	Production efficiency[c]
	Total	Domestic production		All	Sawnwood	WBP[b]	Paper & paperboard			
1900–1950	112.0	106.4	1.7	94.6	90.1	0.6	3.9	1.9	19.1	83.2%
1951–1955	66.5	65.5	0.3	26.5	24.4	0.2	1.8	0.6	40.3	39.7%
1956–1960	151.6	150.6	0.3	55.0	50.2	0.6	4.2	0.3	96.9	36.2%
1961–1965	138.3	134.7	0.1	42.7	37.8	0.6	4.3	0.0	95.7	30.8%
1966–1970	151.7	149.4	0.2	51.6	44.4	1.0	6.2	0.2	100.3	33.9%
1971–1975	197.1	193.9	0.6	53.0	43.1	1.7	8.2	0.6	144.6	26.8%
1976–1980	232.9	225.7	1.6	63.6	49.3	3.3	11.0	2.8	170.8	27.1%
1981–1985	279.5	249.5	3.1	85.5	63.1	6.8	15.6	5.6	197.1	30.2%
1986–1990	298.8	264.7	4.6	99.3	60.6	12.5	26.2	9.9	204.2	32.7%
1991–1995	288.1	268.7	10.9	155.7	76.7	33.3	45.7	27.5	143.3	52.1%
1996–2000	344.7	288.4	18.7	206.5	71.1	66.9	68.5	54.1	156.9	56.8%
2001–2005	508.4	280.8	46.6	400.7	50.9	179.8	170.0	49.6	154.3	72.2%
2006–2010	921.2	512.5	125.6	943.4	126.9	438.0	378.5	−4.7[d]	103.4	90.1%
2011–2015	1822.9	1223.4	229.6	1857.4	256.9	1021.1	579.5	16.8	195.0	90.5%

[a] The difference between the total virgin fiber consumption and the domestic production is the net import of virgin fiber in the period.
[b] Wood-based panels that include plywood, fiberboard, particleboard, and other wood-based panels.
[c] Calculated by dividing "The production of all finished HWP" (column 5) by the the sum of "total virgin wood consumption (column 2) and the "domestically recycled paper" (column 4).
[d] The negative value indicates export exceeded import.

2016), municipal waste disposal options in China include burning, composting, recycling, or disposed of in landfills and open dumps. Recycling is an important disposal option for retired paper and paper products in China while composting is not used (Cai et al., 2013). Therefore, we assumed all retired HWP in China are either burned or disposed of in landfills or open dumps, while recycling is an additional option for retired paper and paperboard. We obtained the data on recycled paper from Almanacs of China Paper Industry from 1986 to 2015 (CTAPI, 2016). According to Cai et al. (2013), open dumps were used as the main disposal option for municipal waste before 1980; since then, the fraction of disposing municipal waste into landfills has been increasing rapidly. By 2007, most of the municipal waste has been either disposed of in landfills or combusted, while open dumps were no longer used (Zhang, 2016). Therefore, for estimating retired HWP disposed of by these disposal options, we assumed that from 1900 to 1980, all retired HWP were disposed of in open dumps. Since 1981, the fraction discarded in landfills increased by 3% each year to 0.70 in 2015, and the portion combusted increased by 3% from 0.01 in 2004 to 0.30 in 2015; As a result, the fraction of HWP discarded in open dumps decreased to zero by 2007 (Table D.1, Appendix D).

Recycled HWP, assumed to be used as wood fiber input to produce new HWP in the next year, continue to store carbon. Burning waste HWP immediately releases the carbon they contained into the atmosphere, while HWP disposed of in landfills and open dumps decompose over time. HWP disposed of in open dumps decompose under aerobic conditions, producing primarily CO_2. We assumed that under anaerobic conditions, 50% of the HWP in landfills decompose over time (treated as a decomposing carbon component), while another half do not decompose or decompose extremely slowly (treated as a permanent carbon component) (IPCC, 2006b; Skog, 2008). The decomposing HWP carbon component in landfills, as well as the carbon stocks of retired HWP disposed of in open dumps, can be estimated using the first-order decay method illustrated by Eq. (5) using half-lives presented in Table C.1 (Appendix C).

$$C_{Decompose}(t) = e^{-k} \times C_{Decompose}(t-1) + \left[\frac{1-e^{-k}}{k}\right] \times f_{Decompose} \times f_{Landfill}(t-1) \times C_{Discard}(t-1) \quad (5)$$

where $C_{Decompose}(t)$ is the cumulative carbon stock of the decomposing carbon component of retired HWP in landfills in year t; $f_{Landfill}(t-1)$ is the fraction of waste HWP disposed of in landfills in year $t-1$; $f_{Decompose}$ is the fraction of HWP in landfills that decompose anaerobically (0.5). The cumulative carbon stock of the permanent carbon component in landfills, $C_{Permannent}(t)$, can be estimated using Eq. (6):

$$C_{Permanent}(t) = \sum_{t} (1 - f_{Decompose}) \times f_{Landfill}(t-1) \times C_{Discard}(t-1) \quad (6)$$

The total HWP carbon stock in landfills is the sum of the two components calculated using Eqs. (5) and (6). The carbon released to the atmosphere through HWP decomposition in landfills can be estimated using Eq. (7):

$$R_{Landfill}(t) = [C_{landfill}(t-1) + f_{Landfill}(t) \cdot C_{Discard}(t-1)] - C_{landfill}(t) \quad (7)$$

where $R_{Landfill}$ is the annual HWP carbon emissions in landfills; $C_{Landfill}$is the cumulative carbon stock of HWP in landfills in year $t-1$.

Of the total landfills carbon emissions, 50% is released as CO_2, and another half as methane (IPCC, 2006b). Landfill methane collection is currently not applied in China (Cai et al., 2013; Zhang, 2016). We assumed 10% of the methane is oxidized when it reaches the top layer of soil covering the waste in municipal landfills (IPCC, 2006b). To include all of the above factors, the CO_2-eq of annual methane emissions in landfills can be estimated via Eq. (8):

$$E_{Methane}(t) = 0.5 \times R_{Landfill}(t) \times \frac{16}{12} \times 28 \times 0.9 \quad (8)$$

where $E_{Methane}(t)$ is the CO_2-eq of methane emission in year t; 0.5 indicates half of the carbon is released as methane; 16/12 is the molar ratio for converting from mass of C to mass of methane; 28 is the global warming potential of methane; 0.9 reflects that 10% of the methane is oxidized.

3. Results

3.1. Wood carbon balance during harvested wood products manufacturing

Wood carbon balance (cumulative carbon stocks results after considering existing HWP carbon stocks, and periodic carbon input to and emissions from HWP carbon pool) during HWP manufacture in 1900–2015 is presented in Table 1. Since 1950, the consumption of virgin wood fiber and the domestically recycled paper increased continuously, and increased significantly since 2000. The total wood fiber input to HWP manufacture in the period 2011–2015 were estimated to be 2052.5 Tg CO_2-eq, of which virgin wood fiber input and domestically recycled paper accounted for 1822.9 and 229.6 Tg CO_2-eq, respectively. Compared to the period 1951–1955, the total input fiber increased by 30 times. A total of 3597.2 Tg CO_2-eq of virgin wood fiber and 417.5 Tg CO_2-eq of the domestically recycled paper were consumed by China's HWP manufacture industry in recent two decades, which accounted for 58.9% and 90.5% of the total wood fiber and domestically recycled paper during 1900–2015, respectively, reflecting the recent rapid development of HWP industries in China. Net imports of virgin wood fiber accounted for only 5.5% of the total virgin wood fiber consumed by HWP industries in the period 1996–2000, but its share increased rapidly, and increased to 29.2% in the period 2011–2015. As a result, the production of HWP in China increased at an annual rate of 3.8% in 1951–1990, 7.6% in 1991–2000, and 15.8% in 2001–2015. In general, the production of all of the three categories of HWP (sawnwood, wood-based panels, and paper and paperboard) increased since 1950. In particular, the wood-based panel production increased by 15.2% each year in the period 1951–2015, much faster than that of sawnwood (4.0%) and paper and paperboard (10.1%). In the last 5-year period in Table 1, the carbon contained by wood-based panels accounts for 55.0% of that the total carbon stored in all HWP produced in this period.

The production efficiency showed a U-shape in the last 65 years, the period we focused on in this study (Table 1): the efficiency decreases continuously from 39.7% in 1951–1955 to 27.1% in 1976–1980, possibly due to the increased production of sawnwood that accounted for more than 80% of the carbon of all finished HWP in China in this period. Sawnwood has a lower utilization rate of wood fiber compared to other finished HWP, while mill residue from sawnwood production were largely not used to produce other finished HWP prior to 1980 (SFA, 2016). In addition, large amounts of industrial roundwood were used to manufacture crossties and pitwood before 1980 in China (SFA, 1986). But these two products were not included in our analysis, because no production data are available, contributing to underestimating the production efficiency. Since 1980, production of paper and paperboard increased by 12% annually in China, while production of wood-based panels increased at an annual rate of 18.7% since 1996. These HWP consumed increasingly more mill residue over time, resulting in a dramatically improved overall HWP production efficiency in the last 2–3 decades of the study period. From 2006 to 2010, the overall HWP production efficiency of China's HWP industries stabilized at 90%, a high level that is about the same as in Canada (Chen et al., 2014) and the United States (Heath et al., 2011).

The total consumption of input wood fiber, and thus the total production and consumption of HWP in China in the period from 1900 to 1950 are very small, compared to that of the last 65 years of the study period (Table 1). Hence, we'll focus on the period 1950–2015, for which we also have better data to support the analysis, to report and discuss HWP carbon stocks and emissions.

Total carbon contained by waste mill residue increased from 19.1 Tg CO_2-eq in 1950 to 1822.2 Tg CO_2-eq in 2015. Of the total 1822.2 Tg CO_2-eq, 62.6 Tg CO_2-eq was combusted and released, 1181.7 Tg CO_2-eq was disposed of in stockpiles, and 577.9 Tg CO_2-eq was discarded in landfills (Table 2). Although stockpiles were used to dispose 64.9% of the total waste mill residue, the amount of mill residue disposed of in stockpiles decreased since 1980. And started from 2010, no more mill residue is disposed of in stockpiles. As a result, the cumulative mill residue carbon stocks in stockpiles decreased since 1990, when the emissions from decomposing wood materials in stockpiles exceeded the newly discarded mill residue. In contrast, the carbon stock of mill residue in landfills accounted for approximately 85% of the total carbon contained by the waste mill residue that disposed of in landfills in the last three decades of the study period. However, the methane emissions generated by decomposing mill residue in the landfills increased at an annual rate of 12.0% from 1985 to 2015, resulting in the reduced mill residue carbon stocks. Overall, the total carbon stock of waste mill residues remained increasing from 1950 to 2015. However, methane emissions from industrial landfills increased rapidly, and since 2000, the increases in methane emissions have exceeded the increases of mill residue carbon stocks, resulting in reduced net carbon stocks in industrial landfills.

The net import of finished HWP into China accounted for a much smaller fraction in total HWP consumption, compared to the HWP manufactured using domestically produced virgin wood fiber (Table 1). And thus, the net imports of finished HWP had minor effects on the life-cycle carbon stocks and emissions analysis for HWP manufacturing in China.

3.2. Carbon stocks of in-use harvested wood products

Cumulative carbon stocks of in-use HWP from 1950 to 2015 are showed in Table 3. The carbon stocks of HWP in use increased continuously in the 65 years, and increased sharply in the last 10 years of this period. Among major HWP categories, the carbon stocks of in-use wood-based panel products significantly increased from 0.8 Tg CO_2-eq in 1950 to 1458.7 Tg CO_2-eq in 2015 (an average annual increase rate of 12.1%). In comparison, the carbon stocks of in-use paper and paperboard increased at an annual rate of 10.4% from 1950 to 2015, while the carbon stocks of in-use sawnwood and the direct use of roundwood increased at slower rates.

By 2015, the cumulative carbon stock of HWP in use was estimated to be 3284.2 Tg CO_2-eq, of which solid HWP (sawnwood, and wood-based panels, and direct use of roundwood) accounted for 2861.2 Tg CO_2-eq, while in-use paper and paperboard stored 298.3 Tg CO_2-eq. Among different solid HWP types, in-use wood-based panels accounted for 51.0% of the 2861.2 Tg CO_2-eq of carbon storage, while the in-use sawnwood and the direct-use of roundwood were estimated to retain 837.7 and 564.8 Tg CO_2-eq, respectively.

3.3. Post-service harvested wood products carbon stocks and emissions

The increases of HWP consumption in China resulted in the increases of HWP retired from their end uses (Table 4). The total carbon contained by all of the HWP retired from 1900 to 2015 was estimated to be 2501.5 Tg CO_2-eq, of which 443.9 Tg CO_2-eq was recycled, 159.4 Tg CO_2-eq was immediately released through combustion, 885.7 Tg CO_2-eq was disposed of in open dumps, and 1012.5 Tg CO_2-eq was discarded in landfills. Because retired HWP discarded into open dumps decreased since 1990, and soon after 2005, open dumps were no longer used as a disposal option for retired HWP, the HWP carbon stocks in open dumps started to decrease since 2000. Carbon stocks of HWP in landfills increased significantly since 1980 when landfills became a disposal option to discard retired HWP. By 2015, the cumulative carbon stocks of retired HWP in landfills were estimated to be 907.5 Tg CO_2-eq. At the same time, cumulative landfill methane emissions were estimated to be 481.1 Tg CO_2-eq. Overall, the carbon stock of retired HWP increased steadily to 1093.2 Tg CO_2-eq by 2015, while the net carbon stock was estimated to be 659.4 Tg CO_2-eq after considering the methane emissions.

Table 2
Cumulative carbon stocks and emissions (Tg CO_2-eq) of the HWP waste mill residue in China, 1950–2015.

Year	1950	1955	1960	1965	1970	1975	1980	1985	1990	1995	2000	2005	2010	2015
Total Waste	19.1	59.5	156.4	252.1	352.4	497.0	667.9	865.0	1069.2	1212.5	1369.5	1523.8	1627.2	1822.2
Combustion	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	12.3	62.6
Stockpile	19.1	59.5	156.4	252.1	352.4	497.0	667.9	844.2	996.0	1079.7	1142.3	1180.3	1181.7	1181.7
Carbon stock	10.1	46.9	130.3	197.0	253.7	342.0	438.2	521.8	564.1	534.8	492.3	433.9	352.9	286.1
Landfill	0.0	0.0	0.0	0.0	0.0	0.0	0.0	20.8	73.2	132.8	227.2	342.5	433.2	577.9
Methane emission[a]	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9	10.0	32.2	67.5	126.6	203.5	298.7
Carbon stock	0.0	0.0	0.0	0.0	0.0	0.0	0.0	20.6	70.5	124.0	208.7	307.9	377.7	496.4
Net carbon stock[b]	0.0	0.0	0.0	0.0	0.0	0.0	0.0	19.7	61.5	95.0	147.8	193.7	194.1	227.0
Total carbon stock	10.1	46.9	130.3	197.0	253.7	342.0	438.2	542.3	634.6	658.8	701.1	741.8	730.6	782.4
Total net carbon stock[b]	10.1	46.9	130.3	197.0	253.7	342.0	438.2	541.5	625.6	629.8	640.2	627.6	547.0	513.1

[a] Methane emissions were converted to CO_2-eq after considering the 28 times of global warming potential of this GHG gas.
[b] Net carbon stocks were calculated by deducting the methane emissions from total cumulative carbon stocks.

Table 3
Cumulative carbon stocks (Tg CO_2-eq) of in-use harvested wood products (HWP) in China, 1950–2015.

Year	1950	1955	1960	1965	1970	1975	1980	1985	1990	1995	2000	2005	2010	2015
Total carbon stock	192.3	229.2	292.4	339.8	394.6	449.0	513.6	596.7	687.3	834.8	1009.7	1294.8	1949.9	3284.2
Solid HWP	191.6	227.5	289.0	336.3	389.9	442.2	503.0	581.6	663.6	788.7	929.9	1137.7	1662.3	2861.2
Direct use	127.8	145.6	165.2	186.6	210.3	236.4	265.2	297.2	332.7	372.1	407.7	433.1	497.6	564.8
Sawnwood	62.9	80.8	122.2	147.5	176.7	201.4	230.4	270.4	304.1	352.3	394.3	426.9	543.9	837.7
Wood-based panels[a]	0.8	1.1	1.7	2.2	2.9	4.4	7.4	14.0	26.9	64.4	127.9	277.7	620.7	1458.7
Paper & paperboard	0.7	1.7	3.3	3.5	4.7	6.9	10.6	15.1	23.6	46.1	79.8	157.1	287.6	423.0

[a] Wood-based panels include plywood, fiberboard, particleboard, and other wood-based panels.

3.4. Carbon budget of China's harvested wood products, 1900–2015

The overall carbon budget for HWP consumed by China is illustrated in Fig. 2. From 1900 to 2015, the total carbon contained by all the virgin wood fiber consumed by China was estimated to be 7884 Tg CO_2-eq, of which 6484 and 1400 Tg CO_2-eq were virgin wood fiber produced domestically and from net imports, respectively. In addition, paper recycled in this period contained a total of 444 Tg CO_2-eq of carbon, which was used to produce new paper and paperboard. Of the 6484 Tg CO_2-eq of the domestically produced wood fiber, 1485 Tg CO_2-eq was consumed by direct uses (e.g., rural log house constructions), and 884 Tg CO_2-eq was fuelwood with its carbon directly released after the wood was burned. Our results show that 5959 Tg CO_2-eq of wood materials entered the HWP manufacturing processes, including 3194 Tg CO_2-eq of industrial roundwood, 921 Tg CO_2-eq of logging residue, and 444 Tg CO_2-eq of recycled paper. Of all the wood materials entered HWP manufacturing processes, 1822 Tg CO_2-eq of the carbon was converted to waste mill residue during HWP manufacturing, with the remaining 4135 Tg CO_2-eq of carbon converted to various finished HWP. We estimated 63 Tg CO_2-eq of the 1822 Tg CO_2-eq of carbon contained by the waste mill residue, was released to atmosphere through combustion, and the remaining 1182 Tg CO_2-eq was discarded into stockpiles where 896 Tg CO_2 of carbon was released through decomposition. Industrial landfills were used to discard 578 Tg CO_2-eq of the waste mill residues, where wood decomposition resulted in 52 Tg CO_2 emissions and 299 Tg CO_2-eq of methane emissions. Overall, the cumulative net carbon stock of the mill residue discarded in open dumps and industrial landfills were estimated to be 513 Tg CO_2-eq.

In addition to the 6504 Tg CO_2-eq of domestically produced end-use HWP from 1900 to 2015, 165 Tg CO_2-eq of net import of finished HWP was consumed by China. Of the total 6669 Tg CO_2-eq contained by all HWP consumed in China that were allocated to various end uses, 3284 Tg CO_2-eq was retained by in-use HWP in 2015, 884 Tg CO_2-eq was released into atmosphere through fuelwood combustion, and 2502 Tg CO_2-eq was contained by retired HWP. Of the 2502 Tg CO_2-eq of carbon contained by the discarded HWP, 444 Tg CO_2-eq of recycled paper and paperboard was recycled, 159 Tg CO_2-eq was burned without energy recovery, and the remaining was disposed of in open dumps (886 Tg CO_2-eq) and landfills (1013 Tg CO_2-eq). By 2015, discarded HWP in landfills and open dumps retained 659 Tg CO_2-eq of carbon, while

Table 4
Cumulative carbon stocks and emissions (Tg CO_2-eq) of retired HWP by disposal option, 1950–2015.

Year	1950	1955	1960	1965	1970	1975	1980	1985	1990	1995	2000	2005	2010	2015
Retired HWP[a]	173.4	209.7	253.2	305.7	366.6	436.9	518.1	614.5	731.7	877.2	1077.2	1357.7	1799.3	2501.5
Recycling	1.7	2.0	2.2	2.4	2.6	3.1	4.7	7.8	12.5	23.4	42.1	88.7	214.3	443.9
Combustion	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.4	36.9	159.4
Open dump	171.8	207.7	251.0	303.4	364.0	433.7	513.4	597.2	679.8	756.4	829.5	881.9	885.7	885.7
Carbon stock	94.7	110.4	129.9	153.9	180.7	210.6	243.6	273.4	295.5	306.7	311.2	293.6	234.6	185.7
Landfill	0.0	0.0	0.0	0.0	0.0	0.0	0.0	9.4	39.4	97.4	205.6	384.8	662.5	1012.5
Methane emission	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	5.4	21.1	57.2	130.0	263.7	481.1
Carbon stock	0.0	0.0	0.0	0.0	0.0	0.0	0.0	9.3	38.2	92.8	193.1	356.4	604.9	907.5
Net carbon stock	0.0	0.0	0.0	0.0	0.0	0.0	0.0	8.8	33.3	73.8	141.5	239.1	367.0	473.7
Carbon stock	94.7	110.4	129.9	153.9	180.7	210.6	243.6	282.6	333.7	399.5	504.2	650.0	839.5	1093.2
Net carbon stock	94.7	110.4	129.9	153.9	180.7	210.6	243.6	282.2	328.8	380.5	452.7	532.8	601.7	659.4

[a] Total carbon contained by all retired HWP from 1900 to the particular reporting year, which was divided by disposal option (recycling, combustion, open dumps, and landfills).

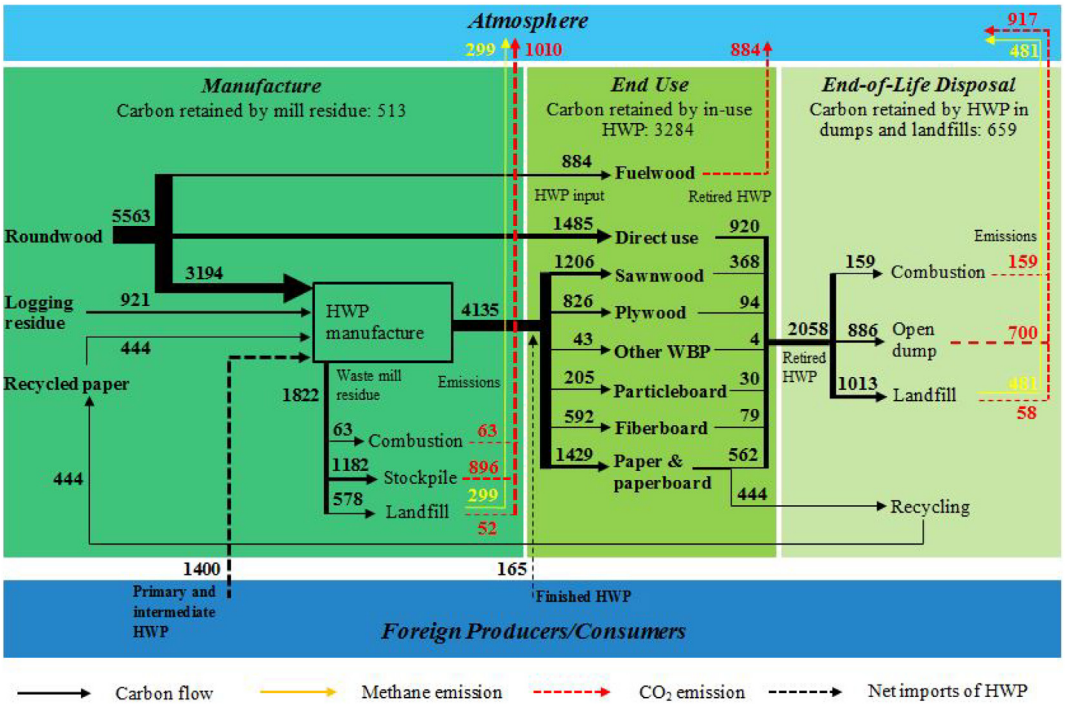

Fig. 2. Harvested wood products (HWP) carbon balance in China, 1900–2015. The wood fiber input, including roundwood, logging residue, net imports of primary, intermediate and finished HWP, was estimated for the period 1900–2015. The sum of life-cycle carbon stocks and emissions (when the global warming potential of methane is not considered) is equal to the total carbon contained by wood fiber input to the HWP life-cycle carbon system. Numbers on the left-hand side of a HWP life stage (production, end use, and end-of-life disposal) represent the carbon inflows to the life stage, the numbers on the right-hand represent the carbon outflows from the life stage and the differences of carbon inflows and total outflows are the net carbon stocks of the HWP life stage. WBP: wood-based panels.

decomposition released 1398 Tg CO_2-eq of carbon back to the atmosphere (917 Tg CO_2 emissions and 481 Tg CO_2-eq of methane emissions). The total carbon emissions from 1900 to 2015, including emissions from combustion of fuelwood, mill residue, and retired HWP, and the decomposition of mill residue and retired HWP disposed of in landfills and open dumps, were estimated to be 3591 Tg CO_2-eq, of which burning fuelwood, combustion and decomposition of mill residue, and combustion and decomposition of retired HWP accounted for 24.6%, 36.5%, and 38.9%.

Overall, the net cumulative HWP carbon stocks in China, calculated as the sum of the carbon stocks contained by mill residue disposed of and HWP in use and in landfills, minus landfill methane emissions produced from decomposing mill residue and retired HWP in landfills, increased steadily from 297 Tg CO_2-eq in 1950 to 4456 Tg CO_2-eq in 2015(Fig. 3). Of the 4456 Tg CO_2-eq of carbon storage, in-use HWP contained 3284 Tg CO_2-eq, and HWP and mill residues in waste disposal sites retained 1172 Tg CO_2-eq.

4. Discussion

The management of HWP uses and post-service disposal can be considered as an extension of forest management. Hence, maximizing the post-harvest carbon stocks and minimizing the associated emissions can enhance the climate change mitigation potential of HWP produced from harvesting sustainably managed forests. Our results suggested that in the recent decades, China's HWP production efficiency has greatly

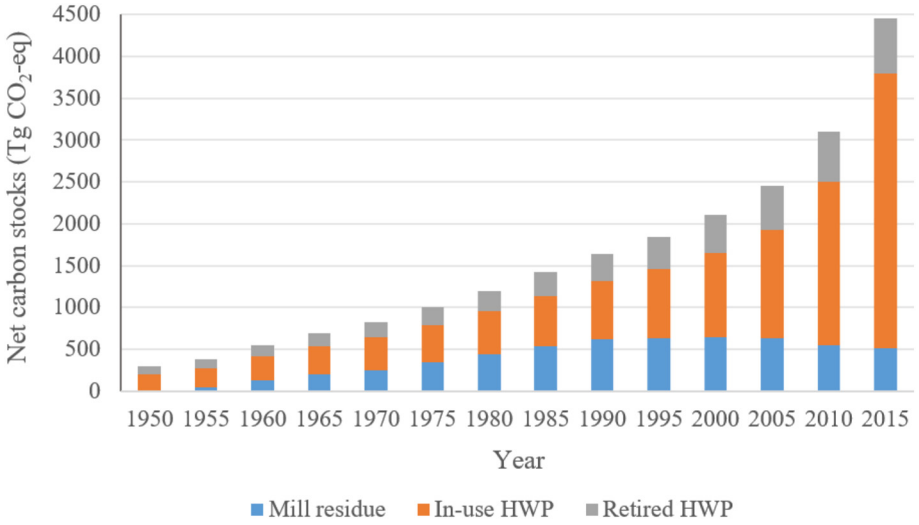

Fig. 3. Cumulative net carbon stocks of harvested wood products (HWP) in China in the period 1950–2015 (teragrams of carbon dioxide-equivalent, or Tg CO_2-eq): Mill residue – net carbon stocks of mill residue disposed of in industrial landfills; In-use HWP – carbon stocks of in-use HWP; Retired HWP – net carbon stocks of retired HWP disposed at waste disposal sites.

improved to 90%, a level that is as almost the same as that in United States and Canada, and is likely difficult to further improve (Chen et al., 2014; Skog, 2008; Heath et al., 2011). Improved HWP production efficiency has manifold effects in enhancing HWP carbon stocks and reducing associated emissions, especially because China is one of the largest HWP producers in the world (SFA, 2016; Yang et al., 2010; Zhang et al., 2016). First, improved production efficiency indicates less mill residue produced, which can become immediate emission if burned, or produce emissions over time through decomposition if disposed of in dumps and landfills. Second, more HWP are produced due to improved production efficiency to store carbon for various periods of time, depending on their end uses. Third, additional HWP produced from improved production efficiency can contribute to reduction of GHG emissions by substituting for more GHG-intensive materials (Chen et al., 2018; Kalt et al., 2016; Kayo et al., 2014). And fourth, through improving HWP production efficiency, forest harvest can be reduced, which is particularly important for a country such as China, because its forest coverage, wood volume per hectare, and mean forest age are all below the global averages (FAO, 2010).

The total HWP carbon stock in China was estimated to be 4456 Tg CO_2-eq in 2015, of which carbon stored by in-use HWP accounted for 73.7% or 3284 Tg CO_2-eq (Fig. 3). The carbon stock we estimated for in-use HWP in China in 2015 is 31.6% greater than the 2496 Tg CO_2-eq estimated by Yang and Zhang (2016). Yang and Zhang (2016) used production and trade data obtained from FAOSTAT, while the data we used was mainly obtained from China Forestry Statistical Yearbooks and Almanacs of China paper Industry. We consider the China-specific data we used are more accurate, compared with the FAOSTAT data that often contain estimations by the Food and Agricultural Organization of the United Nations (FAO, 2010). In addition, we included direct use of industrial roundwood that was not considered in Yang and Zhang (2016). This additional end-use HWP retained 565 Tg CO_2-eq of carbon in 2015, representing 71.7% of the 788 Tg CO_2-eq difference between our estimate and that by Yang and Zhang (2016). Our analysis suggested that the remaining 38.3% of the difference in carbon stocks resulted from difference in the data used by the two studies: FAOSTAT statistics might have underestimated HWP production in China (pers. comm., Kenneth E. Skog, USDA Forest Service; Joseph Buongiorno and Shushuai Zhu, University of Wisconsin-Madison).

Using more HWP in end uses that have longer service lives is an important approach to increase the life-cycle carbon stocks, because the HWP carbon can be stored by these products until they retire. Increasing recycling of retired HWP can enhance in-use HWP carbon stocks and reducing emissions from discarded HWP. Because most of the in-use HWP carbon stocks in 2015 were retained by in-use wood-based panels (Fig. 3), recycling and reuse of wood-based panels may significantly increase the total HWP carbon stocks and reduce emissions.

Our estimation indicated that 57.3% of the retired paper and paperboard was recycled in 2015, which is about 10% lower than that in Canada (Chen et al., 2014), United States (Heath et al., 2011; Skog, 2008), and 30% lower than other Asian countries such as South Korea and Japan (SFA, 2016). Thus, there is room to increase recycling of paper and paper products in China. Paper products contained the most of the carbon compared to other HWP types (Fig. 2). However, the very short half-life and faster decomposition of paper, compared to solid HWP, resulted in the smaller carbon stocks contained by paper and paper products in use and in landfills, and more landfill emissions from decomposition. Thus, increasing recycling of waste paper by fully implementing a classified garbage collection system promoted in China can be a critical measure for China to reduce GHG emissions from paper products (CTAPI, 2016; Zhang, 2016).

Methane emissions from waste management sector are one of the major sources of atmospheric GHG globally (IPCC, 2006b). From 1980 to 2015, 299 and 481 Tg CO_2-eq of methane emissions were generated through wood decomposition in industrial and municipal landfills in China. Disposing waste HWP in municipal landfills has been one of the main options in the recent decades in China (Table D.1, Appendix D), which resulted in large amounts of methane emissions from decomposing HWP in landfills and greatly reduced the climate change mitigation potential of HWP in China. At a certain point of time, the emissions from landfills can exceed the input of waste HWP into landfills, and the landfill can become of net source of carbon (Chen et al., 2014). However, to our best knowledge, energy recovery from burning collected landfill methane to generate energy is not yet a practical option in China (Zhang, 2016; Cai et al., 2013). Thus, promoting energy recovery from using waste HWP and collecting landfill methane can be another option that China should implement to enhance forestry sector's GHG mitigation contribution. Waste-to-electricity pathway has been selected to recover energy from municipal waste in China, resulting in rapid increases in the number of waste combustion power plants constructed since 2010 (Zhang, 2016).

A few limitations exist in the present study. First, HWP service lives were largely determined by their end uses. For example, solid HWP used as construction materials can have 60–100 years of half-lives, while those solid HWP used for residential construction maintenance and remodeling often have a much shorter half-life (Chen et al., 2014; Skog, 2008). But due to the lack of data, we defined the end uses as finished HWP rather than breaking down to specific end uses. Hence, the half-lives we used (Table C.1, Appendix C) may result in underestimated the carbon stocks of in-use HWP for HWP used as construction materials, and overestimated the carbon stocks and emissions of these products in landfills. Therefore, investigations on particular HWP end uses in China remains as a future research need for improving HWP life-cycle carbon stocks/emissions assessment. Second, using SCA, it is difficult to quantify the imported industrial roundwood and immediate HWP that are originated from deforestation and illegal harvesting, which often contribute to the increased GHG emissions to the atmosphere. And finally, the simplification of mill residue disposal, due to the lack of data, can result in greater uncertainty in analyzing the carbon stocks/emissions of the HWP consumed in China. Fortunately, this has become a less important factor in the recent decades, when most mill residue has been used to produce pulp, paper, and panel products, and only a very small fraction was either combusted, or disposed of in dumps and landfills.

It is sometime difficult to identify the timber sources of the HWP consumed using the SCA. Thus, special attention needs to be paid to avoid importing harvested wood originating from illegal harvesting. Based on the FAOSTAT data, the top 10 roundwood/HWP exporters to China accounted for 84.0% of China's total HWP import in 2015, among which Russia, Papua New Guinea, Solomon Island, Mozambique, Ukraine, and Nigeria may have certain degree of illegal logging (Dieter, 2009). Policy development for regulating roundwood/HWP import to ensure HWP imported from sustainably managed forests is a critical need for China in the near future.

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Appendix A. Wood and harvested wood products carbon content factors

Table A.1
Carbon conversion factors for estimating the carbon contained by a unit of wood or harvested wood products (HWP). For roundwood, wood chips, and solid HWP, the carbon factors are estimated in megagram of carbon per cubic meter of wood (Mg C/m^3), while for dry pulp, paper, and paperboard, it is in Mg C per tonne of paper or dry pulp (Mg C/t).

Products	Data sources	Value
Roundwood, wood chip	IPCC (2006a)	0.260[a]
Veneer sheet	IPCC (2014c)	0.253
Sawnwood		0.229
Plywood		0.267
Fiberboard		0.315
HDF[b]		0.335
MDF		0.295
LDF		0.075
Particleboard, other WBP		0.229
Pulp, paper & paperboard		0.386

[a] Average of coniferous and deciduous tree species.
[b] HDF, MDF, and LDF are high density, medium density, and low density fiberboard, respectively; WBP are wood-based panels.

Appendix B. Estimates of the data on HWP production and trade from 1900 to the earliest reporting year of the statistical data in China

The specific data sources used in this study have large data gaps in HWP production, import, and export, especially before 1950. We adopted Eq. (B.1) to estimate HWP production from 1900 to the earliest reporting year of the statistical data, assuming HWP produced before 1900 have insignificant effects on carbon stocks and emissions for HWP produced since 1900 (IPCC, 2006a).

$$V_t = V_{t_0} \cdot e^{U \cdot (t - t_0)} \quad \text{(B.1)}$$

where V_t is the estimated production/exports/imports of HWPs in the year t; t_0 ($t_0 > t$) is the earliest statistic year of the data; $U = 0.0217$ (for Asia) is the annual rate of increase (IPCC, 2006a).

Appendix C. Half-lives

Table C.1
Half-lives of harvested wood products (HWP) in use and HWP disposed of at solid waste disposal sites. We used the average HWP half-lives in dry and wet climates of boreal and temperate zone obtained from IPCC (2006b), because most of the territory area of China is in the boreal and temperate zone that each accounts for nearly half of the total territory area (Zhang, 2016).

Category of half-lives	Products	Data ources	Value
Half-life in use	Sawnwood, direct use of roundwood	IPCC (2014c)	35
	Wood-based panels		25
	Paper & paperboard		2
Half-life in landfill	Solid HWP	IPCC (2006a)	29
	Paper & paperboard		15
Half-life in open dump	Solid HWP	Skog (2008)	16.5
	Paper & paperboard		8.25

Appendix D. Fraction of waste harvested wood products disposed of by combustion, open dump, and landfill

Table D.1
Fractions of waste harvested wood products discarded by disposal options, modified based on China Circular Economy Yearbooks (Zhang, 2016) and Cai et al. (2013).

Year	Combustion	Landfill	Open dump	Year	Combustion	Landfill	Open dump
1901–1980	0.00	0.00	1.00	1998	0.00	0.59	0.41
1981	0.00	0.03	0.97	1999	0.00	0.63	0.37
1982	0.00	0.07	0.93	2000	0.00	0.66	0.34
1983	0.00	0.10	0.90	2001	0.00	0.69	0.31
1984	0.00	0.13	0.87	2002	0.00	0.72	0.28

(continued on next page)

Table D.1 (*continued*)

Year	Combustion	Landfill	Open dump	Year	Combustion	Landfill	Open dump
1985	0.00	0.16	0.84	2003	0.00	0.76	0.24
1986	0.00	0.20	0.80	2004	0.01	0.80	0.19
1987	0.00	0.23	0.77	2005	0.04	0.83	0.13
1988	0.00	0.26	0.74	2006	0.07	0.87	0.07
1989	0.00	0.30	0.70	2007	0.10	0.90	0.00
1990	0.00	0.33	0.67	2008	0.10	0.90	0.00
1991	0.00	0.36	0.64	2009	0.13	0.87	0.00
1992	0.00	0.40	0.60	2010	0.14	0.86	0.00
1993	0.00	0.43	0.57	2011	0.16	0.84	0.00
1994	0.00	0.46	0.54	2012	0.21	0.79	0.00
1995	0.00	0.49	0.51	2013	0.27	0.73	0.00
1996	0.00	0.53	0.47	2014	0.30	0.70	0.00
1997	0.00	0.56	0.44	2015	0.30	0.70	0.00

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